SECTION B. PROCEDURES FOR THE MONITORING OF BENTHIC COMMUNITIES AROUND POINT-SOURCE DISCHARGES

H L Rees and C Heip

1 INTRODUCTION

This review of procedures for the monitoring of benthic communities around discharge points is based on recommendations of the ICES Benthos Ecology Working Group. These appeared as guidelines in the 1988 report of the ICES Advisory Committee on Marine Pollution (ICES, 1989), and subsequently in the Monitoring Manual of the Oslo and Paris Commissions (OSPARCOM, 1990a).

Though similar principles apply in the planning of discharge studies both intertidally and subtidally, attention is focussed on the latter, where more sophisticated - and costly - remote sampling methods are usually required. We have also concentrated on soft-sediment communities, which are those most commonly encountered in pollution studies and are generally amenable to routine quantitative study. However, alternatives are also considered.

There are a number of key reference works dealing with the general issues of benthic sampling and analytical methods which should be consulted prior to any benthic survey, notably Holme and McIntyre (1984) and Baker and Wolff (1987). These cover the full range of habitats likely to be encountered in monitoring programmes. On a more specific level, examples of recently produced guidelines for impact studies at the sea bed around particular types of point-source inputs (which follow principles similar to those adopted here) include Rees *et al.* (1990) for UK sewage sludge disposal sites, and OSPARCOM (1990b) for North Sea platforms.

The core of most routine monitoring programmes will involve the study of structural properties of benthic communities (e.g., numbers of species and individuals, 'diversity'), in parallel with relevant physical and chemical measurements of the receiving environment. While there is a case for direct study of the effects of waste discharges on benthic processes (e.g., energy flow), further refinements in methodology are needed before routine applications can be advocated (see Section 5 'Analytical methods for measuring community responses', below).

Gray *et al.* (1980) reviewed the attributes of various components of the marine biota in the context of field monitoring of pollution effects. We concur that the benthic macrofauna, i.e., animals living within or in close association with the sea bed and which are retained on 1 mm (or 0.5 mm) mesh sieves, continue to offer the most suitable target in routine monitoring at the community level, for the reasons outlined in Section A, above.

It should be noted that there is increasing interest in the use of meiofauna as a monitoring tool. Many members of this group - conventionally separated from the macrofauna at a mesh size of 0.5 mm - have the advantage (in terms of pollution studies) of having no pelagic larval stage, and individuals are in intimate contact with the pore water by virtue of their small size. Less field sampling effort is required than for the macrofauna, but their main disadvantage lies in the need for a high level of taxonomic expertise (see Section D).

As further research is done on meiofauna, it is to be expected that additional cost-effective options will become available for routine monitoring. Microfaunal analysis, however, is not sufficiently developed at this stage to provide standard procedures.

2 DESIGN AND IMPLEMENTATION OF FIELD SAMPLING PROGRAMMES

In the case of new waste arisings, it is essential that benthic monitoring programmes commence **before** the onset of discharges, to allow the identification of any effects by direct comparisons of pre- and post-disposal data. However, it is recognised (see below) that a modified strategy will be required in cases where discharges pre-date impact assessments.

2.1 STAGE 1: Desk Study

The starting point in any benthic survey must involve an appraisal of the environment at and around the (proposed) receiving area, entailing:

1) a review of the literature on the biology, physiography and hydrography;

2) a review of human impacts - past and present - including port/harbour construction, offshore structures, dredging/dredgings disposal, other point-source or diffuse waste inputs (e.g., nutrients, possible eutrophication), fishing practices;

3) a review of uses - present and predicted - including waste disposal, commercial fishing, shipping, recreation; and

4) an assessment of potential impact (scale and intensity) of the discharge in the water column and/or at the sea bed.

The outcome of this review will determine the degree of monitoring effort, if any, which will be required to meet the objectives of waste disposal management. In the event of **any uncertainty** regarding the predicted outcome of waste discharge, a programme of biological monitoring is advocated. In stages 2-5 below, it is assumed that there are grounds for anticipating some interaction between discharge products and the sea bed biota.

On average, a macrofauna sample will take 1-2 days to analyse by an experienced individual, but there may be considerable variation outside this range (see Section 4 'Laboratory processing of samples', below). Without prior knowledge of the sampling area, it will be difficult at this stage to assign costs to samples; this will depend on the outcome of stage 2, below. Other important cost considerations include site accessibility - and therefore ship-time and size - and time allocated to data processing/reporting.

2.2 STAGE 2: Planning a Sampling Programme

If existing local information on the benthic biota and supporting habitat is adequate, then the programme may proceed to stage 4, below. If not, then systematic sampling over an appropriate spatial scale is required, using a grid of stations.

The area covered will depend on tidal and residual movements, as well as the nature and quantity of the waste. As a general rule, the **minimum** area covered should enclose the zone of initial sea-bed impact, if known, or alternatively an area defined by at least one tidal excursion from the discharge point. Sampling should also encompass nearby depositional areas, if any, within which there may be a possibility of accumulation of any persistent contaminants in the longer term.

The nature of the sea bed will determine the most effective type of sampling gear. In hitherto unworked areas, a pilot survey using a range of devices will be required in order to resolve any uncertainty regarding the nature of the substrate. This will have the additional benefit of allowing an assessment to be made of the relative importance of the biota living within, on or just above the sea bed. Deployment of still and video cameras, either remotely or by divers, can provide useful data during this phase of sampling (see, e.g., Holme, 1984; George *et al.*, 1985; also Rhoads and Germano, 1986, and O'Connor *et al.*, 1989, for application of a remotely operated sediment profile camera).

In areas of rough ground which are unsuitable for grabs, samples of the biota within or on the sea bed may be obtained by dredging, but usually at the cost of accurate quantification. Alternative methods, such as the use of video or still photography, may be necessary for routine monitoring of such areas (see Section 3 'Field sampling methods', below).

For soft substrates, a grab of standard design conforming with the Day or Van Veen type can be used (see Eleftheriou and Holme, 1984, and Rumohr, 1990, for descriptions; also Section 3 'Field sampling methods', below). These should sample a standard area of 0.1 m^2, should allow access - on retrieval - to the surface sediment to allow sub-sampling, and should have stainless steel buckets to minimise the risk of trace metal contamination, if such determinations are to be made. An estimate of the retained sediment volume should be routinely made; volumes of less than 5 litres on muddy ground, and less than 2 $^1/_2$ litres on hard-packed sandy ground, would normally be discarded.

A sieve mesh of 1 mm can be recommended for field extraction of samples of the benthos from sediments in this type of survey, prior to fixation and return to the laboratory, though local circumstances may occasionally dictate the use of coarser or finer meshes. The implications of mesh size for sample processing and interpretation are dealt with by Eleftheriou and Holme (1984), Bishop and Hartley (1986), Hartley *et al.* (1987), Bachelet (1990), and stage 4, below.

The primary aim of this initial survey is to describe the distribution of the benthos, and to relate this, as far as possible, to habitat type. It should also be possible to identify any major impacts of existing discharges. For soft substrates, samples of the benthic biota should be accompanied by sediment sub-samples for analysis of particle size and a range of physical or chemical contaminant tracers appropriate to the outfall(s) under consideration.

The importance of integrating physical, chemical, and biological approaches to sampling must be strongly emphasised, since changes in the biota near to waste discharges invariably provide only circumstantial evidence for effects. Such evidence can be considerably strengthened by a knowledge of waste transport pathways and the distribution of contaminants, as well as natural environmental variability, though absolute proof will rarely be attainable without recourse to follow-up laboratory investigation.

Because of this descriptive emphasis, and the need for adequate spatial coverage, the information content of such surveys can be maximised - relative to the resources available - by sampling singly at several stations, rather than repetitively at a reduced number.

2.3 STAGE 3: Analysis and Interpretation of Data

At this stage, analysis will be required in order to identify patterns in the spatial survey data, and to allow selection of stations for regular follow-up monitoring (see below). A variety of methods may be employed, either for the analysis of spatial or temporal trends, and these are outlined in Section 5 'Analytical methods for measuring community responses', below.

2.4 STAGE 4: Rationalisation of Sampling Design for Regular Monitoring

The number of sampling stations will largely be governed by spatial heterogeneity at the sea bed, predicted dispersal pathways, and cost considerations. Replicate sampling at a minimum frequency of three per site or stratum may be recommended to allow statistical comparisons between stations in space and/or time. Additionally, the choice of mesh size will have a considerable bearing on sampling accuracy. Because the costs of benthos surveys are usually a function of laboratory processing time, the decisions taken concerning numbers of stations and replicates, along with the timing and frequency of sampling, will be critical to cost-effectiveness.

Local circumstances and, especially, the nature of the waste will dictate the sampling effort required. The simplest case which can be envisaged is the disposal of an inert solid waste with minimum dispersion, the immediate effect of which is the elimination of all biota; clearly the sampling effort required in order to delineate such an effect will be minimal. This is very different from the case of a complex effluent of uncertain composition, where the early onset of any adverse change may be of interest. Here, for example, it may be necessary to use finer sieve meshes (0.5 mm) for extraction of the benthos, along with increased replication, to facilitate the detection of subtle effects.

Important considerations at this stage are the efficiency with which taxa and individuals are sampled, and the proportion of animals retained on different mesh sizes. Upon these will depend the number of samples required to achieve a particular level of precision, which will in turn depend on the survey objectives.

These can be investigated during a pilot survey, or at selected sites on the first occasion at which this sampling stage is implemented. It should be remembered that the cost of collecting additional samples in the field is usually small in comparison to that of laboratory processing, so that those which eventually turn out to be surplus to requirements can be discarded.

A detailed account of these aspects of sampling design is given in McIntyre *et al.* (1984) and Hartley and Dicks (1987). Other relevant sources of information include Andrew and Mapstone (1987), Bros and Cowell (1987), Caswell and Weinberg (1986), Cuff and Coleman (1979; see also Green, 1980, and Cuff, 1980), Green (1979, 1984), Hurlbert (1984), Kingston and Riddle (1989), Millard and Lettenmaier (1986), Saila *et al.* (1976), Skalski and McKenzie (1982), Stewart-Oaten *et al.* (1986), and Walker *et al.* (1979).

In the case of new or existing discharges, a judgement is required as to the degree and spatial extent of degradation of the habitat, if any, which can be accepted. This represents a logical progression from stages 1 to 3, above. Subsequent monitoring will then have as its primary aim the establishment that there is no worsening trend in intensity or extent of impact with time. Clearly, the facility to detect and quantify such changes will be determined both by the adopted sampling strategy and the chosen measures(s) of biological response (see Section 5 'Analytical methods for measuring community responses', below).

It will be noted that, while the emphasis is on the monitoring of temporal trends, the incorporation of a **spatial** element is necessary. In its simplest form, a stage 4 strategy will involve sampling at two sites which are comparable in all respects save for the influence of the discharge. However, in practise, local heterogeneity will invariably demand a greater sampling effort.

The objective at this stage can, therefore, be met by selecting a limited suite of stations which represent those areas of interest for regular follow-up monitoring. These may include zones of waste impact identified from the physical, chemical and/or biological data generated from stage 2 (above), or locations deemed to be potentially at risk.

It should be remembered that the sensitivity of different analytical measures of contamination may vary widely. The presence of detectable levels of contamination does not necessarily imply a biological effect and *vice versa*. Such factors must be considered in weighting the contribution of contaminant data to sampling design.

The sampling design should aim to minimise the influence of extraneous environmental variability; this may be relatively straightforward if, for example, the main dispersal pathway is aligned with depth contours, and environmental conditions along this line are similar. More complicated situations may arise, e.g., where dispersal is across depth contours; in shallow water, this is often accompanied by gradients of substrate type caused by the natural sorting processes of wave and tidal action.

A further complication near to urban or industrial areas is the potentially confounding influence of other discharges in the vicinity. In both these cases, sampling should be designed so as to adequately represent the major strata identified from the descriptive survey.

As a general rule, the frequency of sampling will be greater at the onset of a discharge, in order to allow for: 1) any uncertainty in predicted impact, and 2) stabilisation of impact (intensity and extent) with time. However, many studies of the responses of benthic communities to discharges suggest that only exceptionally will there be a need to sample at a frequency of more than once per year.

Initially, annual sampling at the same time each year is therefore adequate. Ideally, sampling should be conducted within a period (often spring) which will avoid seasonal maxima in larval recruitment, since the transient presence of many of the latter may obscure quantitative trends in adult populations which have been exposed to discharges for longer periods. This can have an additional benefit, since the identification of juvenile stages is often problematic and can add significantly to processing time. The frequency of sampling may subsequently be reduced if there is evidence of stability in the response to the contaminant(s).

The choice of mesh size, as well as timing, will in many cases affect the proportion of juveniles and adults in samples (e.g., Rees, 1984; Bachelet, 1990), and these factors must be taken into account when interpreting the results from regular surveys. At one extreme, it may be noted that small size at settling and slow growth of some macrofauna species may result in failure to recruit to even the finest (0.5 mm) sieve mesh commonly in use within the one- to two-year period that might elapse between surveys (see Buchanan *et al.*, 1986).

2.5 STAGE 5: Establishment of Routine

This will involve adherence to standard protocols for sampling and analysis, and these must include continued monitoring of the relevant physical and chemical parameters. However, some flexibility must be allowed for, e.g., in response to changes in the quantity of waste discharged. The continued validity of the rationalised sampling design should be checked by periodically repeated stage 2 grid surveys.

3 FIELD SAMPLING METHODS

Box-core samplers (such as the Reineck box-core, see Eleftheriou and Holme, 1984) have the potential advantage over grabs of digging deeper into bottom substrates, and creating less disturbance at the sediment surface as a result of a frontal pressure wave which has been associated with the latter (see Eleftheriou and Holme, 1984, and Hartley and Dicks, 1987). However, because of their size and weight, they require larger vessels and relatively calm conditions for their efficient deployment, and so are less versatile than grabs.

Remote grab samplers of the Day or Van Veen type can, therefore, be recommended for routine monitoring of soft sediments. Non-standard equipment should not be deployed. If comparisons are to be made between data sets obtained by different gear, then calibration of performance against recommended designs will be necessary (Rumohr, 1990).

Regarding vertical distribution within sediments, most studies show that the majority of benthic organisms occur in the surface 5-10 cm, and will be adequately sampled by grab. However, the distribution of biomass may be different, in that older individuals, especially of bivalves, may live at depths considerably greater than this. Such occurrences, and their significance to the outcome of monitoring programmes, can only be tested by comparisons of the results of grab and deeper-penetrating core samplers.

Protocols for field sampling, including extraction of the benthos on sieves, and preservation, are given in Eleftheriou and Holme (1984), Hartley and Dicks (1987), Hartley *et al.* (1987), and Rumohr (1990).

Areas of hard ground (e.g., rocks, coarse gravel) present particular problems for quantitative sampling at the community level, as there may be considerable uncertainty as to the sampling efficiency of dredging devices. However, this can be overcome in areas accessible to divers. For example, non-destructive quantitative assessment of the fauna and flora of rock faces has been carried out over several years using stereo photography (Lundalv and Christie, 1986).

Also in shallow subtidal rocky areas, the fauna inhabiting kelp (seaweed) holdfasts, collected by divers, has been used as a monitoring tool (see Moore, 1973). Recent reviews of survey approaches for both inter- and sub-tidal rocky habitats are given by Gamble (1984), Baker and Crothers (1987), and Hiscock (1987).

Monitoring strategies in such localities may benefit from a consideration of the role of individual resident species (especially sedentary bivalves) as indicators of biological effects and/or contaminant bioaccumulation at the population level (e.g., Rees and Nicholson, 1989).

4 LABORATORY PROCESSING OF SAMPLES

The laboratory undertaking will depend on the objectives of data analysis (see Section 5, below), but typically will require the identification and enumeration of all taxa encountered in preserved samples. In most cases, identification to the level of species can be achieved by reference to standard taxonomic keys. The time - and therefore cost - required to achieve this will vary considerably according to:

1) the expertise and continuity of staff;

2) previous knowledge of the area. Familiarity gained from initial surveys, and/or access to historical reference collections of taxa encountered, can considerably enhance the speed and efficiency of processing;

3) mesh size. In general, the larger the mesh size used in sampling, the easier and faster will be the rate of processing, since juvenile stages and a range of adult species of small size will tend to be omitted. Identification of the former, in particular, can be problematic;

4) the nature of the samples. Samples containing large quantities of residual material in addition to the benthic fauna can create special problems at the sorting stage. For example, fine organic detritus, often found in association with muddy depositional areas, may extend the sorting time to several days, in contrast to a fine sandy sediment, where the separation of sediment from the fauna may be virtually complete at the field sampling stage.

Extraction of the biota in the presence of quantities of inorganic material, such as coarse sand or gravel, can be speeded up by simple decantation, or more sophisticated procedures such as Barnett's fluidised sand bath (see Eleftheriou and Holme, 1984; also Pauly, 1973), provided that thorough checks are made on the efficiency of the procedures.

If sub-sampling is considered to be necessary, it should be remembered that while this may be acceptable for species counts, only exceptionally will this account for the full range of taxa present. Thus, sorting of the entire sample is required, unless this deficiency can be tolerated in subsequent comparisons of the data. Such a compromise can impose severe limitations, especially when making comparisons with studies elsewhere and, if possible, should be avoided.

Details of laboratory procedures are given in Eleftheriou and Holme (1984), Hartley *et al.* (1987), and Rumohr (1990). Regarding biomass determinations, these should be expressed as ash-free dry weight using appropriate conversion factors (Rumohr *et al.*, 1987), or following procedures outlined in the report of a recent ICES intercalibration exercise (Duineveld and Witte, 1987; see also Rumohr, 1990).

It will be appreciated from the above that a major proportion of the resources committed to benthic monitoring programmes will be taken up at the laboratory processing stage. Model sampling strategies must be translated into the reality of local routines, and some compromises are invariably necessary. It is, therefore, imperative that this aspect is taken into account at the survey planning stage and adequate resources allocated.

5 ANALYTICAL METHODS FOR MEASURING COMMUNITY RESPONSES

The purpose of this account is to outline the main classes of techniques available for the analysis of benthos data, rather than to provide an exhaustive literature review, which may be found elsewhere (see, for example, Burd *et al.*, 1990, for recent coverage of this topic). Further details on several of the techniques, together with examples of their application to field data on the macrofauna and meiofauna, can be found in Sections A and D.

5.1 Numerical Analysis of Primary and Derived Variables

5.1.1 Primary variables

Following quantitative sampling, determinations of species composition, **densities**, **weight** and preferably **size**, fulfill a basic requirement of most routine benthic monitoring programmes. Any subsequent rationalisation, e.g., selection of target organisms for single-species studies, or identification to the level of higher taxa only, should proceed only after the establishment of a sound baseline of knowledge of the biota in the receiving area.

The variables of total abundance, number of species, and biomass can be surprisingly robust indicators of environmental changes, and have been shown to respond predictably along organic enrichment gradients (see Pearson and Rosenberg, 1978). Moreover, these are explicable in terms of functional responses of the biota to alterations in the benthic habitat. They may be expressed graphically, or by simple mapping techniques, depending on the sampling design.

5.1.2 Derived variables

The ability to detect gradients or trends using the primary variables is often limited. The application of classical univariate or bivariate statistical techniques may even be misleading. A variety of summary statistics and ordering techniques have been developed which may be used for the further elucidation of structure in the data, and to aid the formulation of hypotheses concerning effects of discharges. They may also provide an objective basis for rationalising subsequent sampling programmes (Clarke and Green, 1988).

It should be remembered that a number of these methods employ sophisticated mathematical techniques, a sound appreciation of which is required for the correct interpretation of the output. Heip *et al.* (1988a) provide a critical review of several commonly used methods for the analyses of marine benthic data. Further details are provided in Section D, in the context of meiofauna investigations.

These techniques take three main forms:

a) graphical displays of species-area or species-abundance relationships

Many different methods exist which are variants of ranked species abundance (RSA) curves and k-dominance curves in which species are ranked according to abundance. Species abundance distributions are frequency distributions with a logarithmic or linear ordinate. When geometric abundance classes are used, a log-normal distribution is often found.

These graphs and distributions are useful tools for presenting the data, but since these are empirical relationships, it is difficult to detect causality when changes occur.

b) diversity

Two aspects are recognised: species richness, i.e., the number of species, and equitability, i.e., the distribution of individuals among the species.

Many different indices have been proposed. A coherent system may be found in the diversity numbers of Hill (1973), which include S (the number of species), H' (the Shannon-Wiener index) and Simpson's index, among others. Diversity indices are useful and may be compared statistically. The different diversity numbers in the Hill series cover different structural aspects and thus more adequately represent overall community structure. However, full species-abundance plots contain more information (Heip *et al.*, 1988b).

c) classification and ordination

These are techniques capable of synthesis and ordering of the data. Classification involves arranging the sites or species into groups (clusters), setting them apart from the members of other groups. Ordination attempts to place sites or species in a space defined by one or more axes in such a way that knowledge of their position relative to the axes conveys the maximum information about them.

Classification and ordination should be standard practice to analyse abundance or biomass data obtained in surveys involving many species and stations. The definition of a standard protocol of data analysis is still awaited (but see Gray *et al.*, 1988; Clarke and Green, 1988). However, the general availability of personal computers and software packages brings these methods within easy reach. A bibliography of selected references on this topic is given at Annex 1.

5.1.3 Conclusions

Many options are available within each of the above categories of data analysis, and no single measure can be recommended as suitable in all cases. Rather, there is merit in the application of a variety of different measures of community structure, a procedure which is facilitated by the wide availability of statistical packages.

5.2 Other Measures of Community Properties

5.2.1 Investigations at higher taxonomic levels

The traditional taxonomic unit in marine pollution investigations is that of the species. Recently, a number of workers have demonstrated that gradients of response can often be adequately described at higher taxonomic levels (Warwick, 1988a,b; Heip *et al.*, 1988a,b; Herman and Heip, 1988). Warwick (1988b) argues that natural variables, such as water depth and substrate type, tend to result in species replacement within taxonomic groups as a result of evolutionary adaptation. However, the advent of man-made pollution is too recent for such adaptation, and hence within-taxon species changes in response to natural variability will tend to be overridden by a uniformity in response at higher taxonomic levels, e.g., at the level of the family or above.

Such an approach could also result in significant reductions in time spent identifying rare species; this might facilitate the processing of more samples, often a practical limitation in pollution surveys.

Presently, validatory work has concentrated largely on the retrospective analysis of data from well-defined pollution gradients. While the technique clearly has promise, a wider programme of testing of its efficiency is required before its routine adoption can be recommended.

5.2.2 Trophic groups

For most species, the predominant feeding mode can be deduced from the literature, and such information may be used as a measure of the response of the benthic fauna to waste discharges. For example, a shift in dominance from suspension-feeding to surface deposit-feeding may indicate excessive turbidity, or increased accumulation of organic matter at the sea bed (e.g., Pearson, 1971).

One limitation which should be borne in mind is that some species have the ability to switch from one feeding mode to another (e.g., Buhr, 1976). Dauer (1984) provides a critique of the utility in impact studies of classifying polychaetes into feeding guilds, as far as current knowledge permits.

5.2.3 Vertical distribution of fauna

Soft sediments present an array of habitats suitable for faunal colonisation both within and on the substrate, including, for smaller-sized species, the interstices between particles. Effects of discharges may be manifested through a reduction in habitat suitability, e.g., due to sediment accretion, or the development of anoxic conditions at depth as a result of excessive organic matter accumulation.

Periodic examination of the vertical distribution of the fauna (e.g., from undisturbed sediment cores), along with measures such as redox potentials, may therefore be useful in assessing the progress of any degradation process within sediments (e.g., Pearson, 1987).

5.2.4 Size spectra

The response of benthic communities to pollution may be expressed in terms of changes in the frequency distribution of body size, e.g., for the macrofauna, smaller 'opportunistic' species are commonly found to predominate near to discharges, at the expense of larger, slow-growing species (Pearson and Rosenberg, 1978; Pearson, 1987; see also Warwick *et al.*, 1986). This approach requires exhaustive treatment of samples if carried out over the full size range of the benthic biota (e.g., Schwinghamer, 1983), and - though a promising area of research - could not be advocated for routine monitoring.

Recently, Warwick (1986) and Warwick *et al.* (1987) have proposed a method of pollution detection based on the relation between ranked abundance and biomass curves of the macrofauna at individual sites (see Section A). The technique may prove to be useful, but presently requires further empirical testing.

5.2.5 Community metabolism

Impairment of community function in response to waste inputs may have consequences for the supply of energy (as food) to higher trophic levels, and may be expressed through changes in, e.g., oxygen uptake, ETS (electron transport activity), ATP concentrations, and heat production.

These may be measured *in situ* or from sediment sub-samples and can be converted into units of carbon or energy flow. As rate functions, they can contribute to models of ecosystem energy flow, and thus have potential application in assessments of assimilative capacity, or vulnerability to contaminant effects (e.g., Pamatmat *et al.*, 1981; Graf *et al.*, 1984; de Wilde *et al.*, 1986).

Presently, methodological problems and uncertainty regarding the accuracy of some of the measured responses preclude their routine application. However, the general approach is considered to have potential value as a monitoring tool, for use in conjunction with traditional studies of community structure.

5.2.6 Annual production calculated from growth and mortality rates

This entails separate examination of each of the dominant species using methodology outlined by Crisp (1984), followed by summation to obtain an estimate of community production. The approach has the advantage of providing data on: 1) the performance (e.g., growth rates) of individual species in proximity to discharges; 2) the 'carrying capacity' of the receiving environment, in energetic terms; and 3) the nature and quantity of biomass available as food for fish.

The main disadvantage is that the sampling and initial analytical effort can be time consuming; at least seasonal sampling is required. It may be possible to overcome this using annual P:B (production:biomass) ratios for species studied elsewhere, but this may not always be valid, since the ratios may vary substantially from one region to another. The approach cannot presently be advocated for routine monitoring programmes, but the potential for future application should improve as the database expands (see Brey, 1988).

6 REFERENCES FOR PART B

Andrew, N.L., and Mapstone, B.D. 1987. Sampling and the description of spatial pattern in marine ecology. Oceanogr. Mar. Biol. Ann. Rev., **25**: 39-90.

Bachelet, G. 1990. The choice of a sieving mesh size in the quantitative assessment of marine macrobenthos: a necessary compromise between aims and constraints. Mar. Environ. Res., **30**: 21-35.

Baker, J.M., and Crothers, J.H. 1987. Intertidal rock, pp. 157-197. *In* Biological surveys of estuaries and coasts. Ed. by J.M. Baker, and W.J. Wolff. Cambridge University Press, Cambridge.

Baker, J.M., and Wolff, W.J. (Eds.) 1987. Biological surveys of estuaries and coasts. Cambridge University Press, Cambridge. 449pp.

Bishop, J.D.D., and Hartley, J.P. 1986. A comparison of the fauna retained on 0.5 mm and 1 mm meshes from benthic samples taken in the Beatrice Oilfield, Moray Firth, Scotland. Proc. Roy. Soc. Edinburgh, **91B**: 247-262.

Brey, T. 1988. Empirical relations between production, P/B ratio, biomass and mean individual weight in macrobenthic invertebrates. Paper submitted to the ICES Benthos Ecology Working Group, Texel, 1988. 35pp.

Bros, W.E., and Cowell, B.C. 1987. A technique for optimizing sample size/replication. J. Exp. Mar. Biol. Ecol., **114**: 63-71.

Buchanan, J.B., Brachi, R., Christie, G., and Moore, J.J. 1986. An analysis of a stable period in the Northumberland benthic fauna - 1973-80. J. Mar. Biol. Ass. UK, **66**: 659-670.

Buhr, K.-J. 1976. Suspension feeding and assimilation efficiency in *Lanice conchilega* (Polychaeta). Mar. Biol., **38**: 373-383.

Burd, B.J., Nemec, A., and Brinkhurst, R.O. 1990. The development and application of analytical methods in benthic marine infaunal studies. Adv. Mar. Biol., **26**: 169-247.

Caswell, H., and Weinberg, J.R. 1986. Sample size and sensitivity in the detection of community impact, pp. 1040-1045. *In* Oceans 86, Conference Record, **3**: Monitoring Strategies Symposium. Inst. of Electrical and Electronic Engineers, New York.

Clarke, K.R., and Green, R.H. 1988. Statistical design and analysis for a 'biological effects' study. Mar. Ecol. Prog. Ser., **46**: 213-226.

Crisp, D.J. 1984. Energy flow measurements, pp. 284-372. *In* Methods for the study of marine benthos. Ed. by N.A. Holme, and A.D. McIntyre. IBP Handbook No. 16 (2nd ed). Blackwell Scientific Publications, Oxford.

Cuff, W. 1980. Comment on optimal survey design: reply. Can. J. Fish. Aquat. Sci., **37**: 297.

Cuff, W., and Coleman, N. 1979. Optimal survey design: lessons from a stratified random sample of macrobenthos. J. Fish. Res. Board Can., **36**: 351-361.

Dauer, D.M. 1984. The use of polychaete feeding guilds as biological variables. Mar. Pollut. Bull., **15**: 301-305.

Duineveld, G.C.A., and Witte, H.J. 1987. Report on an intercalibration exercise on methods for determining ash-free dry weight of macrozoobenthos. Doc. ICES C.M.1987/L:39.

Eleftheriou, A., and Holme, N.A. 1984. Macrofauna techniques, pp. 140-216. *In* Methods for the study of marine benthos. Ed. by N.A. Holme, and A.D. McIntyre. IBP Handbook No. 16 (2nd ed). Blackwell Scientific Publications, Oxford.

Gamble, J.C. 1984. Diving, pp. 99-139. *In* Methods for the study of marine benthos. Ed. by N.A. Holme, and A.D. McIntyre. IBP Handbook No. 16 (2nd ed). Blackwell Scientific Publications, Oxford.

George, J.D., Lythgoe, G.I., and Lythgoe, J.N. (Eds.) 1985. Underwater photography and television for scientists. Clarendon Press, Oxford. 184pp.

Graf, G., Bengtsson, W., Faubel, A., Meyer-Reil, L.-A., Schulz, R., Theede, H., and Thiel, H. 1984. The importance of the spring phytoplankton bloom for the benthic system of Kiel Bight. Rapp. P.-v. Réun. Cons. int. Explor. Mer, **183**: 136-143.

Gray, J.S., Boesch, D., Heip, C., Jones, A.M., Lassig, J., Vanderhorst, R., and Wolfe, D. 1980. The role of ecology in marine pollution monitoring. Rapp. P.-v. Réun. Cons. int. Explor. Mer, **179**: 237-252.

Gray, J.S., Aschan, M., Carr, M.R., Clarke, K.R., Green, R.H., Pearson, T.H., Rosenberg, R., and Warwick, R.M. 1988. Analysis of community attributes of the benthic macrofauna of Frierfjord/Langesundfjord and in a mesocosm experiment. Mar. Ecol. Prog. Ser., **46**: 151-165.

Green, R.H. 1979. Sampling design and statistical methods for environmental biologists. J Wiley, New York. 257pp.

Green, R.H. 1980. Comment on optimal survey design. Can. J. Fish. Aquat. Sci., **37**: 296.

Green, R.H. 1984. Statistical and nonstatistical considerations for environmental studies. Environ. Monit. Assess., **4**: 293-301.

Hartley, J.P., and Dicks, B. 1987. Macrofauna of subtidal sediments using remote sampling, pp. 106-130. *In* Biological surveys of estuaries and coasts. Ed. by J.M. Baker, and W.J. Wolff. Cambridge University Press, Cambridge.

Hartley, J.P., Dicks, B., and Wolff, W.J. 1987. Processing sediment macrofauna samples, pp. 131-139. *In* Biological surveys of estuaries and coasts. Ed. by J.M. Baker, and W.J. Wolff. Cambridge University Press, Cambridge.

Heip, C., Herman, P.M.J., and Soetaert, K. 1988a. Data processing, evaluation and analysis, pp. 197-231. *In* Introduction to the study of meiofauna. Ed. by R.P. Higgins, and H. Thiel. Smithsonian Institution Press, Washington and London.

Heip, C., Warwick, R.M., Carr, M.R., Herman, P.M.J., Huys, R., Smol, N., and Van Holsbeke, K. 1988b. Analysis of community attributes of the benthic meiofauna of Frierfjord/Langesundfjord. Mar. Ecol. Prog. Ser., **46**: 171-180.

Herman, P.M.J., and Heip, C. 1988. On the use of meiofauna in ecological monitoring: who needs taxonomy? Mar. Pollut. Bull., **19**: 665-668.

Hill, M.O. 1973. Diversity and evenness: a unifying notation and its consequences. Ecology, **54**: 427-432.

Hiscock, K. 1987. Subtidal rock and shallow sediments using diving, pp. 198-237. *In* Biological surveys of estuaries and coasts. Ed. by J.M. Baker, and W.J. Wolff. Cambridge University Press, Cambridge.

Holme, N.A. 1984. Photography and television, pp. 66-98. *In* Methods for the study of marine benthos. Ed. by N.A. Holme, and A.D. McIntyre. IBP Handbook No. 16 (2nd ed). Blackwell Scientific Publications, Oxford.

Holme, N.A., and McIntyre, A.D. (Eds.) 1984. Methods for the study of marine benthos. IBP Handbook No. 16 (2nd ed). Blackwell Scientific Publications, Oxford. 387pp.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Monogr., **54**: 187-211.

ICES. 1989. Procedures for the monitoring of benthic communities around point-source discharges, pp. 28-45. *In* Report of the ICES Advisory Committee on Marine Pollution, 1988. ICES Coop. Res. Rep., No. 160.

Kingston, P.F., and Riddle, M.J. 1989. Cost effectiveness of benthic faunal monitoring. Mar. Pollut. Bull., **20**: 490-496.

Lundalv, T., and Christie, H. 1986. Comparative trends and ecological patterns of rocky subtidal communities in the Swedish and Norwegian Skagerrak area. Hydrobiologia, **142**: 71-80.

McIntyre, A.D., and Pearce, J.B. (Eds.) 1980. Biological effects of marine pollution and the problems of monitoring. Rapp. P.-v. Réun. Cons. int. Explor. Mer, No. 179. 346pp.

McIntyre, A.D., Elliott, J.M., and Ellis, D.V. 1984. Introduction: design of sampling programmes, pp. 1-26. *In* Methods for the study of marine benthos. Ed. by N.A. Holme, and A.D. McIntyre. IBP Handbook No. 16 (2nd ed). Blackwell Scientific Publications, Oxford.

Millard, S.P., and Lettenmaier, D.P. 1986. Optimal design of biological sampling programs using the analysis of variance. Estuar. coast. Shelf Sci., **22**: 637-656.

Moore, P.G. 1973. The kelp fauna of northeast Britain. I. Introduction and the physical environment. J. Exp. Mar. Biol. Ecol., **13**: 97-125.

O'Connor, B.D.S., Costelloe, J., Keegan, B.F., and Rhoads, D.C. 1989. The use of REMOTS technology in monitoring coastal enrichment resulting from mariculture. Mar. Pollut. Bull., **20**: 384-390.

OSPARCOM. 1990a. Procedures for the monitoring of benthic communities around point-source discharges. *In* Oslo and Paris Commissions (Principles and Methodology of the Joint Monitoring Programme), Monitoring Manual Part A (A8.2/89-E), 19pp.

OSPARCOM. 1990b. Guidelines for monitoring methods to be used in the vicinity of platforms in the North Sea. *In* Oslo and Paris Commissions (Principles and Methodology of the Joint Monitoring Programme), Monitoring Manual Part D (D1/88-E), 31pp.

Pamatmat, M.M., Graf, G., Bengtsson, W., and Novak, C.S. 1981. Heat production, ATP concentration and electron transport activity of marine sediments. Mar. Ecol. Prog. Ser., **4**: 135-143.

Pauly, D. 1973. Über ein Gerät zur Vorsortierung von Benthosproben. Bericht der Deutschen Wissenschaftlichen Kommission für Meeresforschung, **22**: 458-460.

Pearson, T.H. 1971. Studies on the ecology of the macrobenthic fauna of Lochs Linnhe and Eil, west coast of Scotland. II. Analysis of the macrobenthic fauna by comparison of feeding groups. Symposium d'Arcachon, Vie et Milieu, Suppl. No. 22, **1**: 53-91.

Pearson, T.H. 1987. Benthic ecology in an accumulating sludge disposal site, pp. 195-200. *In* Oceanic processes in marine pollution. **1**: Biological processes and wastes in the ocean. Ed. by J.M. Capuzzo, and D.R. Kester. R E Krieger, Florida, USA.

Pearson, T.H., and Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. Ann. Rev., **16**: 229-311.

Rees, H.L. 1984. A note on mesh selection and sampling efficiency in benthic studies. Mar. Pollut. Bull., **15**: 225-229.

Rees, H.L., and Nicholson, M.D. 1989. Trends in lead levels in the horse-mussel from the western North Sea. Mar. Pollut. Bull., **20**: 86-89.

Rees, H.L., Moore, D.C., Pearson, T.H., Elliott, M., Service, M., Pomfret, J., and Johnson, D. 1990. Procedures for the monitoring of marine benthic communities at UK sewage sludge disposal sites. Dept. Agriculture and Fisheries for Scotland, Scottish Fisheries Information Pamphlet No. 18. 79pp.

Rhoads, D.C., and Germano, J.D. 1986. Interpreting long-term changes in benthic community structure: a new protocol. Hydrobiologia, **142**: 291-308.

Rumohr, H. (Ed.) 1990. Soft-bottom macrofauna: Collection and treatment of samples. ICES Techniques in Marine Environmental Sciences, No. 8. 18pp.

Rumohr, H., Brey, T., and Ankar, S. 1987. A compilation of biometric conversion factors for benthic invertebrates of the Baltic Sea. The Baltic Marine Biologists, Publ. No. 9. 56pp.

Saila, S.B., Pikanowski, R.A., and Vaughan, D.S. 1976. Optimum allocation strategies for sampling benthos in the New York Bight. Estuar. coast. Mar. Sci., **4**: 119-128.

Schwinghamer, P. 1983. Generating ecological hypotheses from biomass spectra using causal analysis: a benthic example. Mar. Ecol. Prog. Ser., **13**: 151-166.

Skalski, J.R., and McKenzie, D.H. 1982. A design for aquatic monitoring programs. J. Environ. Management, **14**: 237-251.

Stewart-Oaten, A., Murdoch, W.W., and Parker, K.R. 1986. Environmental impact assessment: "pseudoreplication" in time? Ecology, **67**: 929-940.

Walker, H.A., Saila, S.B., and Anderson, E.L. 1979. Exploring data structure of New York Bight benthic data using post-collection stratification of samples, and linear discriminant analysis for species composition comparisons. Estuar. coast. Mar. Sci., **9**: 101-120.

Warwick, R.M. 1986. A new method for detecting pollution effects on marine macrobenthic communities. Mar. Biol., **92**: 557-562.

Warwick, R.M. 1988a. Analysis of community attributes of the macrobenthos of Frierfjord/-Langesundfjord at taxonomic levels higher than species. Mar. Ecol. Prog. Ser., **46**: 167-170.

Warwick, R.M. 1988b. The level of taxonomic discrimination required to detect pollution effects on marine benthic communities. Mar. Pollut. Bull., **19**: 259-268.

Warwick, R.M., Collins, J.M., Gee, J.M., and George, C.L. 1986. Species size distributions of benthic and pelagic metazoa: evidence for interaction? Mar. Ecol. Prog. Ser., **34:** 63-68.

Warwick, R.M., Pearson, T.H., and Ruswahyuni. 1987. Detection of pollution effects on marine macrobenthos: further evaluation of the species abundance/biomass method. Mar. Biol., **95:** 193-200.

Wilde, P.A.W.J. de, Berghuis, E.M., and Kok, A. 1986. Biomass and activity of benthic fauna on the Fladen Ground (Northern North Sea). Neth. J. Sea Res., **20:** 313-323.